Comments on

Cropland expansion outpaces agricultural and biofuel policies in the United States

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Lark et al. (2015) present a careful and thorough analysis of land use change (LUC) in the United States between 2008 and 2012, years that saw a marked growth in corn ethanol production (from 9 to 13 billion gallons) (Renewable Fuels Association 2015). To conduct their analysis, they used the Cropland Data Layer (CDL), a product of the United States Department of Agriculture (USDA) National Agricultural Statistics Service (NASS) that aims to characterize how cropland is used each year. As the authors acknowledge, this tool is not intended to measure LUC. To account for steady improvement in the CDL over time that may introduce error, especially in the earlier years of CDL data the study employs, the authors use the National Land Cover Dataset (NLCD) to cross check their assignments of land parcels to certain categories. The authors conclude that cropland in the United States expanded by 2.98 million acres on net with gross land conversion estimated at 7.34 million acres. The authors concluded that 5.7 million acres (77%) of this converted land came from grasslands with the majority of acres coming from planted grasslands and those that had been in pasture or hay. A minority 1.6 million grassland acres were estimated to have been converted from long-standing prairie and range-like locations.

The results of this study call into question the United States Environmental Protection Agency's (USEPA) aggregate compliance approach that aims to limit cropland expansion. This mechanism requires USEPA to check the amount of US cropland against a 2007 baseline annually and work with USDA to identify underlying causes of LUC should the cropland acres in any year exceed that 2007 baseline (402 million acres).

In this response to Lark et al. (2015), we aim to examine the merits and drawbacks of their approach that could lead to inaccuracies in their results, to consider ways to use other data and analyses to assess their conclusions, and to put their conclusions in the context of broader trends.

CDL and NLCD are undoubtedly valuable and high-quality data sets. Nonetheless, while Lark et al. assert that a combination of CDL and NLCD datasets theoretically increases the overall accuracy of their results, they fail to provide statistical proof to support their assertion. The absence of this proof may not mean that the ultimate findings are necessarily incorrect but a thorough CDL (and combined NLCD) error analysis would increase confidence in their results. For example, the accuracy of the CDL varies significantly by state. NASS reviews this and other limitations of the data set, meticulously cataloguing them for users (USDA 2015).

The CDL Super Matrix Accuracy Assessment Normalized for All States reveals that many of the states (North Dakota, South Dakota) the paper identifies as experiencing the largest cropland conversions also exhibit the lowest CDL accuracy levels (Table 1). Moreover, given the critical role grasslands play in Lark et al.'s analysis, data quality pertaining to this land category deserves scrutiny. The accuracy of alfalfa and non-alfalfa hay in the CDL data layers is much lower (less than 50%) than the respective overall accuracy of the CDL layer and a recent detailed analysis of the 2006 NLCD datasets (which are used by Lark et al.) revealed that agriculture gain and loss classes had accuracies below 40% (Wickham et al. 2013). In summary, the low accuracies for grassland type vegetation in both the CDL and the NLDC calls for a detailed discussion of the combined error resulting from the merged use of these datasets in Lark et al.

Table 1. Accuracy of 2012 CDL data (USDA 2014)

Accuracy	Range	North Dakota	South Dakota
Overall	17-92%	55%	35%
USDA Farm Service Agency crops	64-94%	80%	75%

New approaches to characterizing the landscape can improve the quality of datasets used to assess land use and LUC. For example, the USDA is collecting unique, comprehensive, high-resolution visible imagery across agricultural areas on a yearly basis through the National Agriculture Imagery Program (NAIP) with selective coverage dating back to 2002 (USDA FSA 2014). This imagery is often used to ground truth land use with consistently high accuracy (Wickham et al. 2013, Merry et al. 2013).

We utilized NAIP imagery to verify several of the hot spots for cropland conversions identified by Lark et al. The base years for this analysis were, depending on imagery availability, either 2006/2007 and final years 2012/2014. Our analysis therefore covers the period Lark et al. examined. According to Figure 1 in Lark et al., hotspots for conversions occur along the north-central border of North Dakota with South Dakota and south-eastern Iowa. Our preliminary analysis (see Appendix 1) did not show large-scale conversions except for isolated tree removals and minor agricultural conversions. In particular, while conversions of irregularly shaped long-standing and range-like grassland parcels to row crop-type field boundaries can be detected with NAIP imagery, there is no indication that this type of transition has occurred on a large scale in the identified "hot spot" regions. However, land use in these areas is complex and more thorough follow-up analyses are indeed needed. Other findings in Lark et al. also merit deeper analysis. For example, the authors identified a significant role for wheat in land-use change. Common agricultural practice for wheat, however, is to rotate it with grass/pasture and other dryland systems. Lark et al. may have overestimated the amount of land moved permanently from grasslands into wheat.

To put Lark et al.'s conclusions in perspective, it is instructive to consider historical trends in land use. For example, in the 1980s, 350 million acres of land produced crops in the United States. The net increase in agricultural land that Lark et al. (2015) identify is very small compared to this number. (Total acres planted in 2013 were 311 million.) Moreover, urban land has expanded by approximately 15.7 million acres (about 15%) between 2000 and 2010 (United

States Census Bureau 2010). Conversion of native vegetation or cropland to urban lands is essentially irreversible and displaces biological carbon stocks. Changes in agricultural land must be considered alongside changes in other land types such that the primary drivers of land conversion are addressed with priority. Moreover, changes in cropland are also influenced by fuel prices and dietary shifts, not solely by biofuel policy.

One key conclusion that Lark et al. reach is that crop production pushes into lower productivity, marginal lands. This conclusion contradicts Swinton et al.'s (2011) conclusion that despite a 64% increase in profitability from 2006 to 2007-09 for corn, the land area in crop production in the U.S. increased only by 2%; marginal lands were not significantly impacted. Swinton et al. remarked that owners of non-croplands were reluctant to convert this land to agricultural production. Additionally, farmer decisions to convert land rely on more than policy, but on economics, weather conditions and other factors. Lark et al.'s implicit assignment of cropland expansion to biofuel policies is difficult to verify.

Furthermore, Lark et al. assign GHG emissions to the land transitions they estimate based on emission factors from other studies (Gelfand et al. 2011, Fargione et al. 2008). The emission factor from Fargione et al. (2008) that Lark et al. adopted as an upper bound (134 Mg CO_2e ha^{-1}) was based on converting central grasslands to corn. Choosing this single coarse factor will not provide much insight into actual SOC changes that will occur in different regions. Gelfand et al. (2011) developed an emission factor of 68 Mg CO_2e ha^{-1} for grassland converted to no-till soy production. This emission factor includes emissions from decomposing soil and plant carbon during the conversion year, changes in N_2O and CH_4 fluxes, farming activities, a fossil fuel offset credit (for when the biomass is converted to a fuel and consumed in lieu of fossil fuels). It also includes *foregone* carbon sequestration in the soil during the year of conversion and into the future (22 years). Finally, it includes future decomposition of grass that would emit CO_2. Clearly this emission factor encompasses more than merely carbon stock changes and may overestimate carbon stock changes upon conversion of grassland to soy.

Furthermore, Gelfand et al. (2011) base changes in soil carbon accumulation between grassland and cropland on data from the Kellogg Biological Station in Michigan. Yet, a primary area of concern for grassland conversion that Lark et al. identify is the Dakotas. It is important to consider that changes in soil carbon are spatially dependent and influenced by local factors including climate, soil type, and prior land use history (Qin et al. 2015). One indication of whether conditions in Michigan and the Dakotas are alike enough to assume that SOC changes would be similar upon grassland-to-cropland transitions is to examine the agricultural ecological zones (AEZ) in which each state resides. Michigan is located in AEZ 10 (temperate sub-humid) whereas the Dakotas are in AEZs 7 (temperate arid) and 8 (temperate dry semi-arid). These different AEZs indicate notable differences in soil type and climate. One tool to check how SOC changes upon grassland transitions to cropland might be different in these AEZs is to employ modeling as we have done to estimate SOC changes at a county level for key biofuel-relevant land transitions (Dunn et al. 2013, Kwon et al. 2013). Our modeling results, contained within the Carbon Calculator for Land Use Change from Biofuels Production (CCLUB) module of Argonne National Laboratory's Greenhouse gases, Regulated Emissions and Energy use in Transportation

(GREET™) model, show marked differences in emission factors between AEZs 7, 8, and 10 for grassland-to-cropland conversions (Table 2).

Table 2. Emission factors for grassland-to-corn production land transitions in AEZs 7, 8, and 10. No stover is assumed to be harvested. Emission factors for 100-cm soil depth are developed on a 30-year time horizon (Gelfand et al. (2011) use a 22-year time horizon) and assume annual increases in yield based on historical trends. No foregone sequestration or agricultural operations are included. Negative emission factors mean SOC loss.

AEZ	Representative State	Emission factor (Mg CO_2e ha^{-1}) No till	Reduced till
7	Dakotas	70	58
8	Dakotas	45	33
10	Michigan	0.22	-13
11	Kentucky	-28	-41
12	Alabama	-50	-61

Table 2 illustrates that the change in SOC upon grassland transition to cropland is very spatially dependent. Under a no-till scenario in Michigan, one might expect relatively little change in SOC whereas in the Dakotas, SOC could increase when grassland is converted to corn, which was responsible for 27% of land expansion in the Dakotas. Under a reduced-till scenario, however, it is likely SOC would decline if grassland were converted to corn production in AEZ 10. Soils in AEZs 7 and 8, however, could still gain carbon. We have not modeled grassland-to-soy conversions to provide insight into SOC emission factors for that land transition. The key point here is that SOC emission factors should be spatially explicit to engender a level of confidence in estimates of SOC changes and their contribution to overall GHG emissions from LUC.

Examining LUC in the US is an important step in examining potential influences of producing biofuel feedstocks on land use patterns. With Lark et al.'s contribution taken into account, the next step in identifying any concerning shifts in the amount and types of agricultural land and attendant GHG emissions should involve two key aspects. First, the most recent and powerful land use data sources should be used, such as visual imagery contained in the NAIP. Furthermore, should GHG emissions be assigned to LUC, using emission factors that are spatially explicit and take into account land use history, weather, and soil type among other considerations is advisable. While land use is a critical issue, so too is the reduction of GHG emissions from the transportation sector, and biofuels are one key route to this aim. Analyses that highlight potential LUC influences of increased biofuel feedstock production must provide dependable results that can identify LUC hotspots and enable design of approaches that can mitigate LUC in these hotspots and optimize the production of low life cycle GHG-intensity biofuels.

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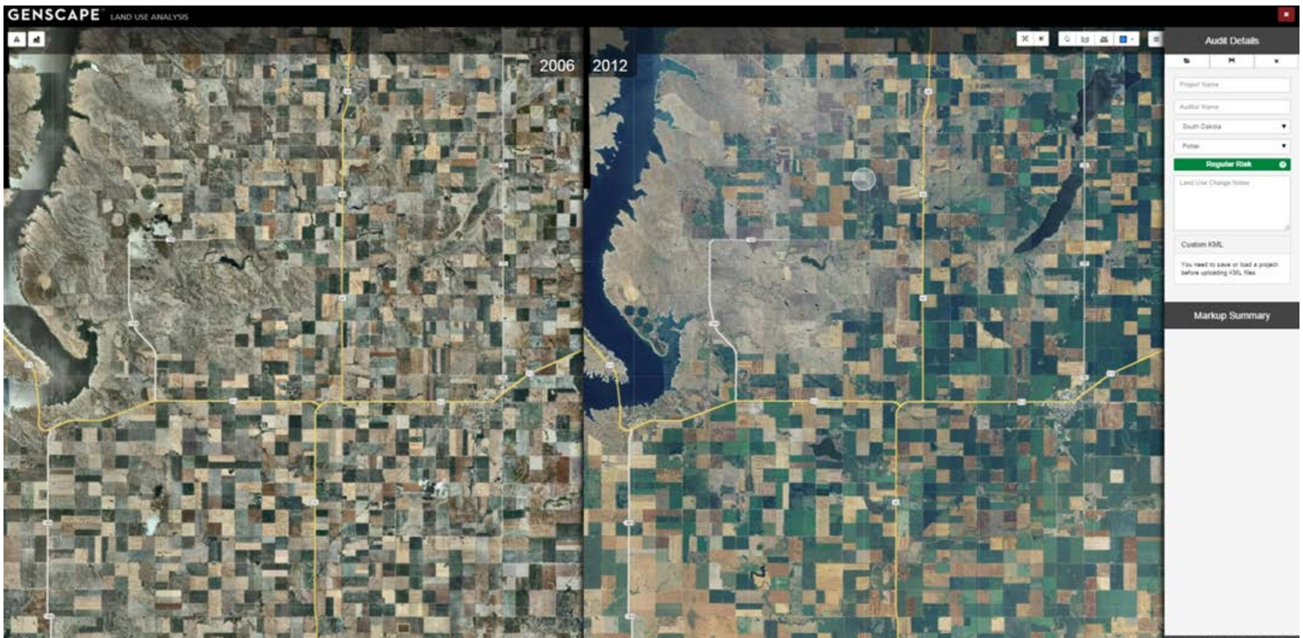
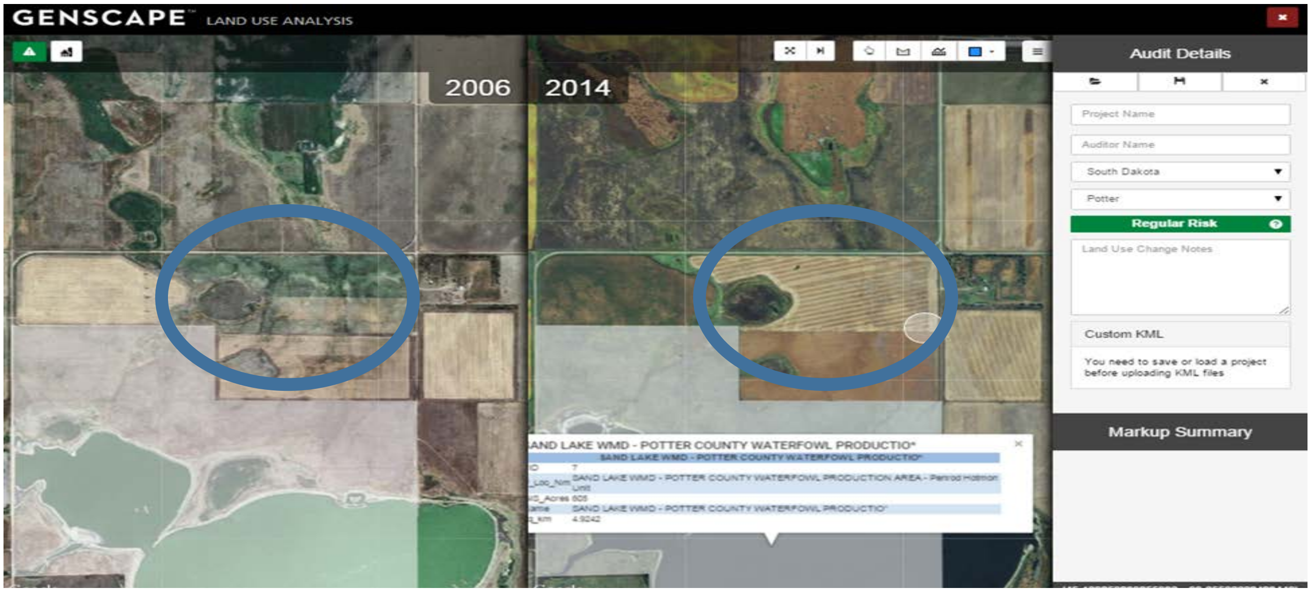
Appendix 1

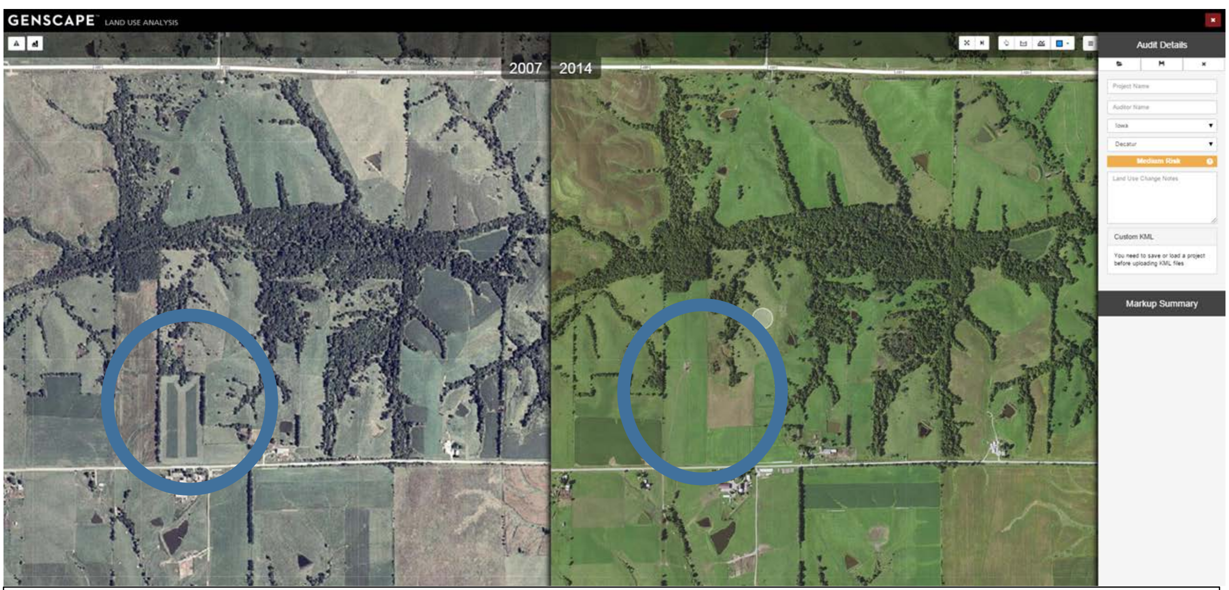
The Genscape-GRAS Land Use Tool utilizes USDA NAIP Imagery (for US) and vegetation signatures (for International Assessments) to facilitate land use change detection and analyses. The left side of the dual viewer interface displays 2006/2007 NAIP imagery (see top corner of image) because 2007 is the base year of interest for land use change under the RFS2. The right side of the viewer displays the most current year of available and updated imagery. The tool can also display protected areas (federal and state level). The tool is hosted at https://apps.genscape.com/Biofuels/LandUse.



Potter County, SD: Large area zoom showing 2006 image (left) vs. 2012 image (right). No conversion.



Potter County, SD: Prairie Pothole. Zoom showing 2006 image (left) vs. 2012 image (right). Detected conversion.



Decatur County, IA: No Conversion. Zoom shows 2007 imagery (left) vs. 2014 imagery (right). Tree buffer reduction but no other conversions.

Ringgold County, IA: Zoom shows 2007 imagery (left) vs. 2014 imagery (right). Some forest parcel reductions.